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Dickson,
B. G.

Landscape level assessment of wildland fires, rare species and exotic plants: implications for fuels management and fire restoration treatments

Landscape level assessment of wildland fires, rare species and exotic plants: implications for fuels management and fire restoration treatments

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Phase III Final Report
(including 2006 Progress Report)

submitted

by

Brett G. Dickson
Ph.D. Graduate Research Assistant
and
Barry R. Noon
Professor, Principal Investigator

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INTRODUCTION

Ponderosa pine (*Pinus ponderosa*) forests of the Southwest were historically characterized by frequent, low severity surface fires (Cooper 1960, Covington and Moore 1994), which created significant spatial and structural heterogeneity among stands on the landscape (Allen et al. 2002, Graham et al. 2004). These surface fires maintained a "patchy" tree distribution, varying in density and size, and mediated the occurrence and return interval of larger, stand-replacing fire events (Moir et al. 1997). However, contemporary increases in surface and canopy fuels have moved the structural characteristics of many ponderosa pine forests from their historical range of variability and toward a more homogeneous condition (Fulé et al. 1997, Swetnam et al. 1999, Sesnie and Bailey 2002). Thus, a century of human-induced manipulation of southwestern fire regimes has resulted in ponderosa pine-dominated ecosystems that are vulnerable to extreme fire events (Cooper 1960, Swetnam et al. 1999). The exceptionally large fires that occurred in the Southwest between 2000 and 2002 attest to this condition and many forest ecologists and fire officials agree that steps must be taken to reduce fuel levels and abate wildland fire risk (Allen et al. 2002, Brown et al. 2004). Land managers are interested in reducing this risk by implementing fuel treatments in progressively larger stands of dry, low elevation ponderosa pine forest. However, there is limited knowledge of how to return forests to a more natural condition and, at the same time, maintain biodiversity and ecological integrity across large areas where fire behavior, fuel quantities, and forest structure have been changed by humans (Covington and Moore 1994, Lyon et al. 2000, Noss et al. 2006).

In recent decades, efforts to restore the structure and function of western forests have accelerated, but debate continues about how best to treat forests and minimize ecological degradation (Covington 2000). Forest restoration methods attempt to establish self-sustaining system processes by manipulating succession and mimicking recovery from natural disturbance (Angermeier and Karr 1994, Covington 2003). Restoration projects in the Southwest are removing large amounts of forest biomass using mechanical thinning and fire treatments in an attempt to create forest structure conditions presumed to exist prior to Euro-American settlement of the region (Covington et al. 1997, Friederici 2003). However, the effects of restoration treatments on various ecosystem attributes are not well understood (Wagner et al. 2000, Kotliar et al. 2002) and effects on sensitive ecosystem components, including wildlife, need to be identified before land managers recommend and implement broad-scale restoration programs (Tiedemann et al. 2000, Allen et al. 2002).

Prescribed fire treatments have been offered as a tool to improve ecosystem health and function (Mutch 1995, Miller and Urban 2000, Kauffman 2004), mitigate wildland fire size and severity (Pollet and Omi 2002, Schoennagel et al. 2004, Finney et al. 2005), and reconcile the goals of fuel treatments and ecological restoration (Allen et al. 2002). These treatments tend to result in low-to-moderate-severity surface fires that reduce ground and ladder fuel quantities, but do little to reduce overstory canopy density (Agee and Skinner 2005). In the Southwest, fire managers typically ignite prescribed fires outside of the summer months, during which natural (i.e., lightning-caused) fire events burn the most area and risk of crown fire is greatest (Swetnam and Betancourt 1990, Dickson et al. 2006). Regardless of season, there remain significant uncertainties about the immediate and long-term environmental consequences of reintroducing fire on plant and animal populations and on the habitats that support biological communities (Tiedemann et al. 2000). Because of these uncertainties, it is important to provide land managers with the information and tools needed to understand and mitigate any adverse effects of prescribed fire treatments on both humans and ecosystem processes.

The ecological impacts of exotic (i.e., non-native or alien) plant species on native communities and biodiversity is recognized as a global problem with severe consequences (see review by Sakai et al. 2001) and many researchers have concluded that exotic species will cause a decline in native plant species and ecosystem diversity (Bock et al. 1986, D'Antonio and Vitousek 1992, Stohlgren et al. 1999). In the U.S., the spread of exotic species poses one of the single greatest threats to both terrestrial and aquatic imperiled species (Wilcove et al.1998, Stein et al. 2000). Moreover, exotic plant species pose a significant threat to native species and community organization in forested ecosystems in the U.S., particularly those systems where the distribution and composition of vegetation has been affected by fire (Hobbs and Huenneke 1992) and fire suppression policies (Cole and Landres 1996, Wilcove and Chen 1998).

For this research, our objectives were to: 1) synthesize forest structural and physiographic habitat attributes at two spatial scales, namely the local (area of < 1 ha) and patch (contiguous area of 1–400 ha) scale, to contrast treatment unit- (~ 400 ha) and site-level (co-located treatment and control units) values of response variables; 2) quantify treatment-induced tree damage and levels of fire severity; and 3) model and predict exotic plant species occurrence and richness as a function of spatial location and ground-based and digitally-derived habitat (environmental) variables, prescribed fire treatment, and fire severity. Specifically, we sought to develop novel statistical models that used spatial covariance to account for spatial autocorrelation in the occurrence and richness patterns of exotic plant species at multiple spatial scales. This modeling approach was developed, in part, to assess if remotely-sensed (digital) data could be used to predict changes in vegetation composition (e.g., exotic plant occurrence and richness), avian species, and ponderosa pine-dominated forest structure attributes in response to large-scale prescribed burns.

Impetus for this final report and our general research approach are described in 01-JV-11221615-186 Phase II Workplan: *Interactions of wildfires and fuels treatments proximate to critical hotspots of rare and exotic species in the Southwest and southern Rocky Mountains*) and 01-JV-11221615-186 Phase I and Phase III Workplans: *Landscape level assessment of wildland fires, rare species and exotic plants: implications for fuels management and fire restoration treatments*). As required by modification #4 to 01-JV-11221615-186, a progress report for the 2006 field season is presented in Appendix A. Project deliverables and related research accomplishments are detailed in Appendix B.

METHODS

Study area

To evaluate exotic plant community response to prescribed fire treatments in ponderosa pine forests on the Mogollon Plateau, we established two study sites on two National Forests in northern Arizona (Figure 1). One site was located on the Kaibab National Forest (KNF), west of Flagstaff and one site was located on the Mogollom Rim Ranger District of the Coconino National Forest (CNF), southeast of Flagstaff. Overstory vegetation on each site was dominated (relative contribution to canopy cover ≥ 80%) by ponderosa pine. Alligator juniper (*Juniperus deppeana*) and Gambel oak (*Quercus gambelii*) were common on the CNF site, but contributed little to overall forest canopy cover. Grasslands occurring on each site were dominated by bunchgrass species, including Arizona fescue (*Festuca arizonica*) and blue grama (*Bouteloua gracilis*). Mean unit elevation was approximately 2,193 m (SD = 58.1, range = 2,122–2,254) and topography among sites was variable (Table 1).

Data collection

Sampling design

In order to establish causal relationships between prescribed fire and changes in the exotic plant community at multiple spatial scales, we used a designed experiment. At each study site, we used a Before-After/Control-Impact (BACI; Green 1979) experimental design to assess each of the response models described below (Figure 2). Each site represented a single replicate consisting of a paired prescribed fire "impact" or "treatment" unit and "control" unit and contained between 80 and 90 permanent sampling stations (hereafter, "points") at which all response variables were measured at 170 total points (Table 1). Each treatment unit was paired with one (CNF) or two (KNF) nearby (1–5 km) control units with similar biotic components, geography, and climate. Specifically, we used a geographic information system (GIS; ArcView® 3.2a and ArcGIS 8.1; ESRI, Redlands, California, USA) and U.S. Geological Survey (USGS) Digital Orthophoto Quads (DOQs; image acquisition occurred 1997-2000) to locate areas for the placement of suitable control units on the landscape (≤ 100 km^2) that included the treatment unit. Total area encompassed by the control unit(s) on each site was roughly equivalent to the total area of the paired treatment unit (396–405 ha). Prior to field work, points were located using methods implemented in a GIS (ArcGIS® v8.3-9.1). We used the GIS to overlay a lattice of 125 × 125-m cells that encompassed each site and randomized the placement of the first cell location. Within the GIS, we systematically placed prospective points at lattice intersections (nodes) when that placement met the following criteria: (1) nodes were $\geq$ 150 m from the unit edge; (2) nodes could be located where stand canopy could be stratified according to average percent cover within a 100-m radius of a point center; and (3) a point could be placed $\geq$ 250 m from its nearest neighboring point. The proportion of points falling within each canopy cover stratum was roughly proportional to the overall representation of each cover stratum on a site. Once points and their corresponding Universal Transverse Mercator (UTM) coordinates were identified in the GIS (zone 12, North American 1983 datum), we used a global positioning system to locate points on the ground at each site and improved positional accuracy by differentially correcting the coordinates for each point using a local base station (implemented in GPS Pathfinder® Office, V3.0, Trimble Navigation Ltd., Sunnyvale, CA, USA).

Prescribed fire treatments

Between September 2003 and May 2004, prescribed fire treatments were implemented by U.S. Forest Service (USFS) District personnel at each of the study sites (Table 2). Because of the large scope and scale of these treatments, it was not possible to randomize the location of each treatment unit, although efforts were made to insure that all sampling was conducted on randomly-placed stations nested within each unit (see below). As outlined in site-specific burn plans, common objectives for each treatment included wildfire threat and hazardous fuels reduction, the maintenance of natural fuelbreaks, and the restoration of natural ecological systems. Desired treatment effects included a 20-90% reduction in trees with a diameter at breast height (dbh) ≤ 8 cm and a 20-50% reduction in trees ≤ 23 cm dbh. Desired environmental conditions included temperatures between -12 and 27° C, relative humidity between 10 and 60%, live fuel moistures between 3 and 35%, and wind speeds ≤ 29 km/h. At each site, fire was applied to a single treatment unit between 247 and 405 ha in extent. Fire personnel used ground-based ignition methods, including head, backing, and/or flanking fires at treatment unit edges and strip-pattern and spot fires at the unit interior. All prescriptions were characterized as broadcast burns with expected fire behaviors of low to moderate intensity. Ignitions were suspended when weather conditions were suboptimal or when average flame lengths exceeded approximately 2 m.

Treatments were completed in the fall of 2003 (CNF) and in the spring of 2004 (KNF; Table 2). Treatment windows ranged from five days (CNF) to five months (KNF) and area burned during each ignition day was variable. In general, a given area was burned only once and during a single ignition day. Although fire personnel attempted to burn all or most of the area encompassed within a prescribed treatment unit, surface fuel characteristics, including continuity and arrangement, and physiographic barriers resulted in incomplete coverage and consumption. Subsequently, fire behavior patterns were spatially heterogeneous and resulted in a mosaic of treatment effects on forest structure. These observed patterns, however, were typical for larger-scale prescribed fire treatments in southwestern ponderosa pine (L. Wadleigh, USFS Region 3 Fire Ecologist, personal communication). In general, among sites, treated areas with substantial amounts of ponderosa pine litter were completely burned and areas composed mostly of grass or oak litter burned incompletely or did not burn (J. Jerman and V. Morfin, USFS District Fire Ecologists, personal communication; personal observation). Typically, smaller (10 hr and 100 hr) fuels were consumed, although 1,000 hr fuels and large (> 30 cm in diameter) logs were only partially (< 50%) consumed (see Saab et al. In Review). On all sites, some torching of ponderosa pine in the smaller diameter classes (e.g., ≤ 8 cm dbh) did occur at the interior of each unit, but mortality was less than expected. Torching of trees in larger diameter classes resulted when burning logs or patches of woody debris generated sufficient heat at the tree bole. Isolated crown fire occurred on each site, but this behavior was less frequent on the CNF site. Ocular estimates of the immediate effects of each treatment suggested that fire behavior was greatest on the KNF and less on the CNF site (see Table 2).

Local-scale habitat attributes

Due to logistical constraints, we summarized local-scale, or "point-level" standing forest structure using ground-based sampling centered on 159 of the 170 points. To characterize these habitat attributes prior to prescribed fire treatments, we placed a 50-m transect line in each of the four cardinal directions and measured the dbh or root-crown diameter (*Juniperus* species) of all live tree species encountered within 5 m of either side of the transect line. To expedite the measurement of smaller (dbh 2.5–23 cm; after Saab et al. In Review) trees, we assigned all stems occurring within 2 m of the transect line to one of six diameter classes. At each point, we estimated total tree density (stems/ha) and basal area (m^2/ha) by combining data for larger (dbh ≥ 23 cm) trees measured within 5 m of each transect line (total sampling area = 0.20 ha) with data for smaller trees measured within 2 m of each transect line (total sampling area = 0.08 ha). For all large live trees measured within the 0.20-ha sampling area, we also estimated mean and variance of tree height from individual measurements made using a clinometer (Suunto Oy, Vantaa, Finland), tree species richness, and the proportion of ponderosa pine stems. We used a digital camera and imaging software (Gap Light Analyzer V2.0) to derive overstory canopy cover values at each point from true-color fisheye photographs (Frazer et al. 1999).

Because large snags were infrequently detected during a pilot effort, we counted the total number of large (dbh $\geq$ 23 cm) snags occurring within 10 m of either side of the transect line and summed these values to estimate the relative number of large snags occurring at each point (total sampling area = 0.40 ha). To quantify snag decay at each point, we first placed each snag into one of four condition classes (after Bull et al. 1997): "1" recently died, had little decay, and retained most of its branches, bark, and top; "2" evidence of decay and was missing some bark, branches, or a portion of the top; "3" extensive decay, bark missing from most or all of the branches, and a broken top; and "4" burnt snag with an outer shell hardened by fire. As a final measure of relative snag condition and decay at each point, we computed a snag decay index by dividing the total number of snags present by the sum of the decay class values recorded for each snag (scaled between 0 and 1, with lower values indicating greater total decay).

Patch-scale habitat attributes

We investigated the relationship between the exotic community and habitat factors beyond the scale of the point (i.e., within a 50-m radius), namely at the patch and landscape (e.g., unit and site; see *Modeling exotic plant response to habitat covariates* below) scales. We considered a patch to be a contiguous area (~ 1-400 ha) surrounding a point location and defined by a vector of forest attribute and physiographic information. To quantify forest habitat characteristics at the patch scale, we used a GIS to derive and map five important physiographic and forest structure attributes on each of the two study sites. We used the Spatial Analyst (ESRI, Redlands, CA, USA) extension to ArcGIS® to derive elevation and slope based on a mosaic of USGS digital elevation models (DEMs). Mean values for elevation and slope were summarized using a neighborhood function (focal mean) and a circular moving window with a 100-m radius. In cooperation with the Forest Ecosystem Restoration Analysis (ForestERA; Sisk et al. 2006) project, we derived three digital maps, or layers, of forest structure using GIS and remote sensing technologies, including canopy cover (0-100%), basal area (m^2/ha), and tree density (individual trees/hectare). We used percent canopy cover as a measure of the total amount of the landscape covered by overstory (tree) canopy foliage. The canopy cover layer was developed from a mosaic of DOQs (image acquisition occurred 1997-2000) and using a new type of advanced exploratory data analysis that relied on empirical information for model training and validation (see Xu et al. 2006). For all trees > 2.54 cm dbh, we used basal area as an index of the total cross-sectional area and tree density as a count of the total number of trees. Using a regression tree methodology and Cubist software (RuleQuest Research Ltd., St. Ives, New South Wales, Australia), the basal area and stem density layers were developed from Landsat 7 Enhanced Thematic Mapper imagery (all scenes acquired between 1999–2000), a Normalized Derived Vegetation Index layer, DEMs, the canopy cover layer described above, and training data from over 600 ground plot locations in northern Arizona and west-central New Mexico (Hampton et al. 2003). Each of the above maps was derived at a 30-m resolution, represented a spatially continuous raster surface in the GIS that encompassed the full extent of each study site, and was projected in UTM coordinates (zone 12, North American 1983 datum) to insure spatial congruence. Each map was then used as one or more habitat covariates included in the statistical models described below.

To identify discrete patch types on each study site, we synthesized the three forest structure attributes using a fuzzy classification algorithm (see Bezdek 1987) implemented with the BoundarySeer® software package (TerraSeer, Inc., Crystal Lake, IL, USA). BoundarySeer® permits the detection, description, and analysis of geographic boundaries in a spatially-explicit, multivariate framework, but has seen only limited application in wildlife ecology (e.g., Hall and Maruca 2001). To identify discrete patches of forest when patch boundaries are necessarily indiscriminate, a fuzzy classification method can be better than traditional, or "hard," classification methods (Brown 1998a and 1998b, Longley et al. 2001). Since the membership of each forest structure attribute within a particular forest patch type is not likely a binary (yes/no) one, fuzzy classification permits the assignment of each attribute to a patch type based on its probabilistic membership. This approach also captured both the spatial and attribute uncertainty surrounding the membership of a given value in any single patch type. For each cell occurring within a unit and a 100-m buffer (to account for edge influences on the clustering algorithm), we identified the value for each of the three forest structure attributes and combined this vector of continuous information as a new, vector-based point data file for analysis in BoundarySeer®. To aid in a more parsimonious interpretation of patch description and patch influence on response variables, we grouped the forest structure data into three classes, or "types." Using output from BoundarySeer®, we used Boolean operators to sort membership probabilities for each attribute into their respective patch type and created a new, categorical raster surface within the GIS. We then implemented a neighborhood function (focal majority) in Spatial Analyst to compute which patch type occurred most frequently within a 100-m radius of each cell and assigned this value to the focal, or center, cell (new covariate = "PatchType"). This new raster surface characterized the spatially-explicit occurrence of patches across each of the units. We computed summary statistics to describe the relative contribution of the original values for each forest structure attribute to each patch type. To quantify patch heterogeneity, or "richness," and the number of different patch types occurring within a 100-m radius of each cell, we used an additional neighborhood function (focal variety) and created a new raster surface (new covariate = "PatchRich"). We associated each survey point with the cell value of the PatchType and PatchRich covariates using an intersect operation in the GIS. We treated the PatchType and PatchRich covariates as ordinal variables in the habitat models described below.

Prescribed fire effects sampling

To assess the effects of prescribed fire on standing forest structure, we used methods consistent with those described by McHugh and Kolb (2003). In collaboration with researchers from Northern Arizona University and the USFS Rocky Mountain Research

Station in Flagstaff, Arizona, we collected information on tree damage for all prescribed fire-affected trees occurring within a single, 10-m radius circular plot (0.03 ha) centered on a point. Tree damage measurements included bole char severity, average maximum bole scorch height, average percent of the bole circumference charred, and average proportional crown scorch. Bole char severity was categorically determined by identifying the position of the bole with the maximum height of charcoal-like bark and then examining the lowest 0.5 m of this position. "Light" char was indicated by only light charring on edges of bark plates. "Moderate" char was indicated by a uniformly black characteristic to the bark plates, except for the innermost depths of prominent fissures. "Heavy" char was indicated when bark plates were deeply charred and their characteristics were not discernable. At each point, we assigned the maximum bole char severity level detected among all fire-damaged trees. We used a clinometer to estimate maximum scorch height at the position on the bole with the maximum height of charcoal-like bark. For each fire-damaged tree, we also estimated the percent of the bole circumference charred at the root crown. To quantify crown scorch and foliage consumption, we used the estimated height of the pre-fire live crown to estimate the total proportion of moderately scorched foliage (brown color) and proportion of foliage that was consumed by the fire.

Exotic plant sampling

Between 2003 and 2004, exotic plants and understory vegetation were sampled on the two study sites (Fowler and Sieg, unpublished data). At each site, a random subset of sampling stations were identified ($40 \leq n \leq 50$) and exotic plant sampling was conducted using plot-based techniques. At each of these sampling stations a single, 80-m transect was placed on the ground using a randomly selected azimuth. The sampling station was the midpoint of this transect line. Beginning at one end of the transect line, five, 5-m radius plots were spaced at 20-m intervals. Within each of these plots, the occurrence of all exotic plants was recorded and used to compute total species richness at each point.

Data analyses

Local-scale habitat

We used variable reduction methods to quantify the multi-dimensional structure of the local-scale forest attribute data and to more parsimoniously fit regression models with uncorrelated predictor variables. We synthesized these data using a factor analysis and extraction by principal components analysis (PCA) to obtain the initial factor solution (Manly 2005). From the resulting variable correlation matrix, we retained only those habitat factors with component score eigenvalues > 1.0 for inference. To simplify interpretation of the variance associated with each retained factor, we used varimax rotation with Kaiser normalization to compute the sums of squared loadings (Manly 2005). For each retained factor, we used the final, or "extracted" communality estimates to describe relative variable importance and considered estimates of 0.50–0.70 and ≥ 0.70 of moderate and most importance, respectively (see McGarigal et al. 2000). This communality estimate is simply the proportion of the variance explained by each local-scale forest structure attribute given the final number of retained factors. For each retained factor, we used the rotated factor loadings to interpret the relative importance of individual variables and used values ≥ |0.50| as a cutoff (Manly 2005). Because the rotation results in the relation of each variable to only a single factor, we used this step to assign ecological meaning to each of the retained habitat factors. For these analyses, we pooled data from all ($n = 278$) points and exported regression scores for the retained habitat factors as independent, local-scale habitat covariates in our point-level habitat models (see below). All analyses were performed using SPSS (V14.0, SPSS Inc., Chicago, IL, USA).

Prescribed fire severity

Zimmerman (2003) defined fire severity as the immediate effects of fire amount, rate, and direction of heat transfer on vegetation, including bole char severity, crown scorch height, and crown consumption. Fire severity can differentially influence the maintenance and arrangement of forest stands (Chappell and Agee 1996, Brown et al. 1999, Knapp and Keeley 2006) and the numerical response of ecological communities (Smucker et al. 2005). To synthesize the multivariate influence of the prescribed fire treatments on these ecosystem components, we developed a fire severity covariate using the five tree damage variables described above. In our analysis, we considered only those points where fire effects were observed and did not include points located on control units. Since one (bole char severity) of the five tree damage variables was categorical, we used a two-step cluster analysis algorithm (Zhang et al. 1996, Chiu et al. 2001; implemented in SPSS) capable of accommodating mixed (categorical and continuous) variable types. For the fire severity data matrix, with variables in the columns and observations at each survey point in the rows, the algorithm consisted of two basic steps: 1) a pre-clustering of records into many small sub-clusters and 2) a clustering of the sub-clusters resulting from the pre-cluster step into a user-defined number of clusters. Both steps use a log-likelihood, probability-based distance measure to maximize the distance between cluster centers. For each step, the distance, d, between two clusters Cx and Cz can be defined as

$$d(x,z) = \xi_x + \xi_z - \xi_{<x,z>}, \qquad (1)$$

where <*x*,*z*> is an index value that represents the new cluster formed by combining clusters x and z, and ξ is the log-likelihood. For $v = x$, z, and $\xi_{<x,z>}$

$$\xi_v = -N_v \left(\sum_{u=1}^{U^A} \frac{1}{2} \log(\hat{\sigma}_u^2 + \hat{\sigma}_{vu}^2) + \sum_{u=1}^{U^B} \hat{E}_{vu} \right), \qquad (2)$$

where N_v is the number of data records in cluster v, U^A is the total number of continuous variables in the procedure, U^B is the total number of categorical variables in the procedure, $\hat{\sigma}_u^2$ is the estimated variance of the u-th continuous variable for all of the data, $\hat{\sigma}_{vu}^2$ is the estimated variance of the u-th continuous variable in cluster v,

$$\hat{E}_{vu} = -\sum_{w=}^{W_u} \frac{N_{vuw}}{N_v} \log \frac{N_{vuw}}{N_v}, \qquad (3)$$

and where W_u is the number of categories for the u-th categorical variable and N_{vuw} is the number of data records in cluster v whose u-th categorical variable takes the w-th category. Using the log-likelihood distance measure, the pre-clustering step reduces the size of the original data matrix by considering distances between all possible pairs of records during a single pass of the data. When a record is read by the algorithm, it uses the decrease in the log-likelihood distance measure to determine whether a record is to be combined with a previously formed precluster, or to initiate a new precluster. This step is also referred to as the construction of a modified cluster feature (CF) tree (Zhang et al. 1996), where each CF tree consists of "nodes" and "leaves," representing a collection of records and a subcluster, respectively. Once this step is complete, all records contained in a precluster are considered as a single subcluster. The second step integrates the more common method of agglomerative hierarchical clustering (see Legendre and Legendre

1998), an iterative procedure that updates the distance between a new cluster and those remaining (<x, z> term in equation (1) above is replaced for either x or z) until a minimum number of clusters is achieved. This step groups all of the subclusters into the desired number of clusters, for which we defined three clusters, or "levels" of fire severity response at each point: low, moderate, and high.

We determined the relative contribution and statistical significance of each tree damage variable in each of the three clusters using a Student's-t (for continuous variables) or chi-square goodness-of-fit (categorical variables) test. Within cluster variable importance was ranked according to the value of the t (absolute) or chi-square statistic. For each variable within a cluster, we assessed statistical significance with respect to its mean value among clusters using $\alpha = 0.05$ and implemented all analyses in SPSS.

Modeling exotic plant response to habitat covariates

I generated estimates for exotic plant richness at the point level for pre- or post-treatment years ($n = 318$ total point-level estimates). I modeled the variance-covariance structure of the point level richness estimates to identify and account for correlation and interdependencies among samples (see below). For each point, j, where $j = 1, 2, \ldots, n$, and n is the 159 points that were sampled, I computed the difference in pre- and post-treatment exotic plant richness as

$$\hat{\Delta}_j = (\hat{R}_B - \hat{R}_U) \text{ or } (\hat{R}_{C2} - \hat{R}_{C1}), \tag{4}$$

and generated estimates for the pre-treatment period (2003) on the unburned treatment (stratum-level estimate = $\hat{R}_U$) and control ($\hat{R}_{C1}$) units and separately for the post-treatment period (2004) on the burned treatment ($\hat{R}_B$) and control ($\hat{R}_{C2}$) units. These point-level difference estimates were used as the response variable in the habitat and fire response models. Specifically, we modeled the temporal and spatial relationship between $\hat{\Delta}_j$ and habitat covariates at the point level. Covariates included: mean elevation, mean slope, PatchType, PatchRich, and the final number of retained local-scale habitat factors. We also included a "treatment" covariate to indicate the control ("0") or treatment unit ("1"). Separately, we modeled the response of exotic plant richness to the four ordinal levels of fire severity (see below). We evaluated three possible functional forms of this response using a simple linear model of the ordinal levels and two transformations: log-normal and second-degree polynomial. For all models, we used mixed-model repeated-measures ANOVA (implemented using PROCMIXED) and models of the general form

$$\mathbf{y} = \mathbf{X\beta} + \mathbf{Zu} + \mathbf{e} \tag{5}$$

where **y** is a vector of observed response values (e.g., fire severity level or $\hat{\Delta}_j$), **X** is a matrix of independent covariate values (the known fixed effects), **β** is a vector of regression parameters (the unknown fixed effects), **Z** is a design matrix for the known random effects (e.g., site), **u** is the vector of unknown random-effect parameters, and **e** is the vector of errors (see Littell et al. 1996). We also modeled the unit-level covariance structure (i.e., positive spatial correlation among points in a given unit). Importantly, this structure permitted me to account for any among-point interdependencies that may have existed in the data. In this case, **e** is assumed to be a vector of correlated experimental errors and its variance is modeled as

$$Var(\mathbf{e}) = \sigma^2 \mathbf{F} \tag{6}$$

where σ^2 is the residual variance among observations, **F** is a covariance matrix with its ab^{th} element defined by $f(d_{ab})$, and where f is a function relating the response variable to a vector of covariates with distance, *d*, between observations *a* and *b* (*a* and *b* = 1, 2,…, *n*, and *n* is the 159 points; see Littell et al. 1996). We used restricted maximum likelihood for covariance parameter estimation and maximum likelihood for fixed effect parameter estimation and to compute AICc (Akaike's Information Criterion corrected for small sample size) values.

We used a multi-model approach to evaluate and predict the relationship between $\hat{\Delta}_j$ and habitat covariates. We used AICc values and Akaike weights to assess model fit and estimated and ranked relative covariate importance by summing the Akaike weights across all (n = 256) possible models in which a given covariate occurred (Burnham and Anderson 2002). Thus,

$$w_i = \frac{\exp(-\frac{1}{2}\Delta_i)}{\sum_{i=1}^{R}\exp(-\frac{1}{2}\Delta_i)} \tag{7}$$

and $\sum w_i = 1.0$ (Burnham and Anderson 2002). After Burnham and Anderson (2002), we computed the unconditional standard error for all model averaged estimates as

$$se(\hat{\bar{\theta}}) = \sum_{i=1}^{R} w_i \sqrt{\hat{\text{var}}(\hat{\theta}_i \mid g_i) + (\hat{\theta}_i - \hat{\bar{\theta}})^2}\,. \tag{8}$$

We considered cumulative Akaike weights (w_+) ≥ 0.50 to be strong evidence for a response to the treatment covariate and proceeded to construct a fire-response model only when we observed this evidence value (Barbieri and Berger 2004). We relied on AICc values to identify the best functional form of the response of each species to levels of fire severity. We used the Tukey-Kramer multiple comparison to test for differences among categories (i.e., levels) of fire severity ($\alpha = 0.05$). We standardized all continuous habitat covariates and UTM coordinates to mean zero and unit variance prior to statistical analysis (Legendre and Legendre 1998) and used Variance Inflation Factors (VIF) to diagnose collinearity among covariates (eliminating covariates with VIF values > 10.0; Neter et al. 1990).

Modeling exotic plant response to forest structure attributes

We also modeled the response of exotic plant richness to the individual forest structure attribute variables at two spatial scales. Separately, we considered the above MIXED model form and used the local- and patch scale canopy closure, stem density, and basal area attributes. In addition to these six covariates, we also included the treatment covariate in our model.

RESULTS

Local-scale habitat variable reduction analysis

Using the local-scale forest structure information, we synthesized nine variables in the habitat factor analysis: tree density, basal area, mean tree height, variation in tree height (SD tree height), percent dominance of ponderosa pine (PIPO dominance), large tree richness, total number of large snags (large snag sum), large snag decay, and overstory canopy closure. The results of the PCA indicated three factors adequately (eigenvalues > 1.00) summarized the nine forest structure attributes and explained 66.6% of the total variance (Table 3). For the three retained habitat factors, final communality estimates indicated that mean tree height, PIPO dominance, large snag decay, large tree richness, and canopy closure were most important, while tree density and basal area were moderately important descriptors of the total variance (Table 4). Large snag sum and SD tree height were less important descriptors (final communality estimates < 0.50). Using the three rotated habitat factor loadings in our interpretation, we considered points with higher values for habitat factor 1 (FAC_1) to be best characterized by increased decay levels in the larger snags, higher stem densities, and greater tree species richness. We considered higher values for habitat factor 2 (FAC_2) to be indicative of points dominated by more open stands of older ponderosa pine and higher values for factor 3 (FAC_3) indicative of points best summarized by high values for canopy closure and basal area.

Patch definition analysis

From the BoundarySeer® outputs and summary statistics for each of the forest structure attributes, we assigned each point to one of three arbitrary forest patch types: "open," "less dense," and "more dense" (Table 5 and Figure 3). Since the concept and composition of a patch is multidimensional (Turner et al. 2001), we used the term "dense" as a descriptor of the amount and relative contribution of horizontal and vertical forest structure attributes. Among sites, patches characterized by a less dense forest structural composition were most common and open areas least common. Within a 100-m radius of a point, patch heterogeneity (i.e., richness) was common. This buffered area around most points intersected two or three patch types, while very few intersected only one.

Prescribed fire effects on habitat analysis

Mean values for scorched and consumed foliage, maximum bole scorch height, and bole scorch circumference increased with the fire severity level defined in the two-step clustering procedure (Table 6). The frequency and proportion of points with low, moderate, and heavy bole char severity also increased with fire severity level. Smaller values for scorched and consumed foliage ranked as most important in defining the low and moderate severity clusters, respectively (Table 7), while large values for bole scorch circumference and bole char severity ranked highest in the high severity cluster. Low and high values for maximum bole scorch height were most important in defining the low and high severity clusters, respectively. Points occurring within the three fire severity clusters were not always characterized by the same category of bole char severity (Table 7). We observed all four levels of fire severity on the KNF site, but did not observe the high level of fire severity on the CNF site.

Exotic plant response to habitat covariates

The difference ($\hat{\Delta}_j$) in pre- and post-treatment exotic plant species richness was not well predicted by the suite of habitat covariates we selected, or at varying spatial scales. There was no clearly best model of exotic plant species richness response to habitat covariates at the point level, although model selection uncertainty was minimal and the best model was > 2 AICc better than the null model (Table 8). The treatment unit and FAC_1 covariates were each in two of the three top models and elevation entered into the model with ΔAICc = 0.00. Cumulative Akaike weights used to assess the relative importance of habitat covariates indicated that the most important covariate was the treatment unit (w_+ = 0.549), followed by elevation and FAC_1 (Table 9). There was less evidence to indicate that the other covariates were important predictors of exotic plant richness. Immediately after prescribed fire, exotic plant species richness decreased in response to treatment (Table 10). Elevation and open or less dense patch types also facilitated a negative response. Because the treatment covariate had a cumulative Akaike weight > 0.50, we modeled exotic plant response as a function of fire severity level (Table 11). Richness was greatest at points with low levels of fire severity, or that were unburned. Moderate and high levels of fire severity resulted in decreased richness, although high levels of fire severity were not observed at points on the CNF treatment unit. Pairwise differences between levels were not statistically significant (all P's > 0.80). The functional form of the response to fire severity level was best described by a null model, although the log normal model was within 2 AICc of the null model (Table 12).

Exotic plant response to forest structure attributes

Consistent with the above results, the difference ($\hat{\Delta}_j$) in pre- and post-treatment exotic plant species richness was not well predicted by forest structure attributes measured at local or patch scales. Model selection uncertainty was high and the null model was < 1 AICc of the top model (Table 13). Notably, local-scale canopy cover entered into most of the models ≤ 2 AICc of the top model. Local-scale canopy cover was the forest structure attribute with the greatest weight of evidence ($w_+ = 0.594$), although there was some support for the treatment covariate ($w_+ = 0.459$; Table 14). There was less support for the patch-scale basal area attribute and little support for the remaining attributes. Immediately after prescribed fire, exotic plant species richness decreased with reduced values of local-scale canopy cover and in response to treatment (Table 15). However, regression coefficients were extremely small and this result should be interpreted with some caution. Patch-scale canopy closure and stem densities also facilitated a negative response. Because the treatment covariate had a cumulative Akaike weight < 0.50, we had insufficient evidence to model the response as a function of fire severity levels.

DISCUSSION

Response of habitat to prescribed fire treatments

Following prescribed fire, we observed substantial spatial heterogeneity in patterns of fire behavior and severity within and among the two treatment units. Such patterns, however, are not unique to this study and have been observed in previous studies of wildland (Agee 1993, Turner et al. 2003, Cocke et al. 2005, Smucker et al. 2005) and prescribed fire (Fulé et al. 2004, Knapp and Keeley 2006). Although each of the treatments implemented as part of this study was designed to maximize coverage of the treatment unit and the consumption of small diameter trees and other hazardous fuels, uniform patterns of fire behavior and severity were minimized by climatic conditions, the arrangement and contiguity of fuels, physiographic barriers, and uneven ignition efforts in space and time. Substantial point-level heterogeneity was also suggested by our regression models of fire severity (not reported herein; see Dickson 2006) – the model with spatial covariance structure was not clearly better than the model without covariance terms. Indeed, these results might be anticipated with any large-scale treatment effort characterized by low- to moderate-intensity prescribed fire and reflect the post-fire heterogeneity in treatment effects typical of such efforts.

The treatments implemented as part of our study were designed to minimize risk of crown fire, and this risk was partly mitigated by using fall and spring ignitions. Typically, fuel moisture and relative humidity are higher and temperature is lower during these seasons, whereas conditions that facilitate increased fire intensity and spread tend to converge in May and June, prior to the summer rainy season (Swetnam and Betancourt 1998). We observed fewer fire-damaged trees and lower levels of fire severity on the CNF, which was treated over five days in September. During this post-monsoonal period, much of the live understory biomass (e.g., grass) was still green and live and dead fuel moisture levels were high. The lower temperatures and higher relative humidity recorded during the KNF treatment, which occurred during winter months, were reflected in the mostly moderate levels of tree damage and severity that we observed on these sites.

The point-level models that we developed to predict exotic plant occurrence and richness did not reveal strong relationships between post-treatment response and the habitat and forests structure predictors we considered. However, our multi-model approach did reveal patterns that can be used to develop new hypotheses about the relationship between exotic plant communities, habitat covariates, scales of measurement, and prescribed fire treatments. For example, compared to all other variables, local-scale canopy closure provided reasonable evidence to suggest exotic plant richness is a function of canopy closure at smaller scales of measurement (i.e., < 50-m radius). To improve the predictive power of our models, we recommend collecting additional biotic and abiotic covariate information (e.g., soil characteristics, distance to road or nearest detection) at this scale (see Davies et al. 2005). Moreover, we would expect considerable improvement in model performance if these data were also collected at the scale at which data on exotic plants were collected. Thus, our results were likely influenced by a lack of covariate information that was spatially or temporally congruent with the exotic plant sampling effort.

The results of both modeling exercises indicated there was evidence for a treatment response by exotic plants, and that the direction of this response was negative. This result may be due, in part, to our assessment of exotic plant response immediately after treatment. Indeed, in more severely burned areas, we might expect the diversity and richness of exotic plants to increase above pre-treatment levels (see Crawford et al. 2001, Griffis et al. 2001). However, research on ponderosa pine-dominated communities by Huisinga et al. (2005) suggests that the small and negative response we observed may also be a function of anthropogenic influence on our sites. In addition to time-since-fire, future models might incorporate the historical context associated with each site and the spatial heterogeneity associated with disturbance.

Although our models explicitly accounted for the covariance structure present in the data, we observed very little spatial structure in point-level estimates of exotic plant species richness. Thus, spatial heterogoneity in the immediate post-treatment occurrence and richness of exotic plants on our sites was probably due to patterns of co-occurrence with factors we did not measure, and that are not presently driven by the intrinsic characteristics of the plants themselves (e.g, their ecological or life-history requisites). Disentangling extrinsic and intrinsic factors at multiple spatial scales (see Davies et al. 2005), coupled with longer-term monitoring, may reveal emergent spatial dependencies.

Management and research implications

In the Southwest, large-scale (e.g., > 10,000 ha) treatments are being considered in order to reduce fire risk adjacent to human communities and restore ponderosa pine forest structure to pre-fire suppression conditions (Allen et al. 2002, Noss et al. 2006). Not surprisingly, the socio-political debate surrounding expanded treatments is polarized

(Mutch 1995, Dombeck et al. 2004) and forest planning efforts are evaluating the merits of various treatment alternatives (Sisk et al. 2006). The degree to which management efforts can mimic historic fire regimes and forest structural conditions using prescribed fire, however, will depend on the timing and frequency of treatment (Sackett et al. 1996, Sackett and Haase 1998). The magnitude of impact of these treatments on forest ecosystem attributes, including exotic plant communities and their potential for invasion, will be dependent on the season within which burning is conducted (Harrington and Sackett 1990, Kerns et al. 2006, Knapp and Keeley 2006).

Because large-scale treatments are expensive and difficult to implement, information on treatment-induced changes to forest structure may be limited or unavailable to most forest managers. If pre-fire forest structure attributes are central in determining post-fire levels of fire severity (Turner et al. 1999, Zimmerman 2003, Cocke et al. 2005, Dickson 2006), then predictive models with covariates that include pre-treatment, or "initial," habitat conditions can be useful to managers interested in understanding the likely outcomes of prescribed fire treatments on forest structure and exotic plant response. These models will be improved by the inclusion of covariate information derived at spatial scales that capture relevant ecosystem processes, such as the factors that influence exotic plant dispersal and establishment.

Non-native plant species can cause undesirable and irreversible change to diverse plant communities and ecosystems and these changes can be enhanced by disturbance, including fire (Crawford et al. 2001, Griffis et al. 2001, Sieg et al. 2003). For example, Crawford et al. (2001) and Griffis et al. (2001) documented significant increases in exotic plant species richness and abundance in response to wildfire in ponderosa pine forests in northern Arizona (but see Huisinga et al. 2005). Similarly, exotic plant species, particularly those considered to be invasive, have the potential to expand dramatically in response to fire used as a tool for restoration or fuels management (Wagner et al. 2000, Croft 2001, Griffis et al. 2001, Sieg et al. 2003). Furthermore, exotic species can respond favorably to mechanical thinning, skidding, slash pile burning, and other restoration or fuels treatment-related activities (Brown 1997, Harrod 2001). Although actions that include removal of exotic species and setting prescribed burns for fire adapted species can be effective tools in the maintenance of biodiversity (Wilcove and Chen 1998, Sieg et al. 2003), fuels and restoration treatment efforts must be informed of the implications of modifying forest conditions to the benefit of exotic plant species and detriment of rare or sensitive community components.

LITERATURE CITED

Agee, J. K. and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83-96 Sp. Iss.

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications 12:1418-1433.

Angermeier, P. L. and J. R. Karr. 1994. Biological integrity versus biological diversity as policy directives. BioScience 44:690-697.

Barbieri, M. M. and J. O. Berger. 2004. Optimal predictive model selection. Annals of Statistics 32:870-897.

Bezdek, J. C. 1987. Some non-standard clustering algorithms. Pages 225-287 *in* P. Legendre and L. Legendre (eds.). Developments in numerical ecology. Springer-Verlag, Berlin, Germany.

Bock, C. E., J. H. Bock, K. L. Jepson, and J. C. Ortega. 1986. Ecological effects of planting African lovegrasses in Arizona. National Geographic Research, 2:456-463.

Brown, D. G. 1998a. Mapping historical forest types in Baraga County Michigan, USA as fuzzy sets. Plant Ecology 134:97-111.

Brown, D. G. 1998b. Classification and boundary vagueness in mapping presettlement forest types. International Journal of Geographical Information Science 12:105-129.

Brown, J. K. 1997. Ecological principles, shifting fire regimes, and management considerations. Pages 185-203 *in* J. K. Brown and J. K. Smith, eds. Wildland fire in ecosystems: effects of fire on flaura. General Technical Report. RMRS-GTR-42-Volume 2. Ogden, UT. USDA Forest Service, Rocky Mountain Research Station.

Brown, P. M., M. R. Kaufmann, and W. D. Shepperd. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology 14:513-532.

Brown, R. T., J. K. Agee, and J. F. Franklin. 2004. Forest restoration and fire: principles in the context of place. Conservation Biology 18:903-912.

Burnham, K. P. and D. R. Anderson. 2002. Model Selection and multi-model inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, NY, USA.

Chappell, C. B. and J. K. Agee. 1996. Fire severity and tree seedling establishment in *Abies magnifica* forests, southern Cascades, Oregon. Ecological Applications 6:628-640.

Chiu, T., D. Fang, J. Chen, Y. Wang, and C. Jeris. 2001. A robust and scalable clustering algorithm for mixed type attributes in large database environment. Pages 263-268 *in* Proceedings of the 7th ACM SIGKDDD international CNFerence on knowledge discovery and data mining. San Francisco, CA, USA.

Cocke, A. E., P. Z. Fulé, and J. E. Crouse. 2005. Comparison of burn severity assessments using Differenced Normalized Burn Ratio and ground data. International Journal of Wildland Fire 14:189-198.

Cole, D. N. and P. B. Landres. 1996. Threats to wilderness ecosystems: impacts and research needs. Ecological Applications, 6:168-184.

Cooper, C. F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecological Monographs 30:129-164.

Covington, W. W. 2000. Helping western forests heal: the prognosis is poor for U.S. forest ecosystems. Nature 408:135-136.

Covington, W. W. 2003. The evolutionary and historical context. Pages 26-47 in P. Friederici (ed.). Ecological restoration of southwestern ponderosa pine forests. Island Press, Washington, D.C., USA.

Covington, W. W. and M. M. Moore. 1994. Southwestern ponderosa forest structure and resource conditions: changes since Euro-American settlement. Journal of Forestry 92:39-47.

Crawford, J. A., C. H. A. Wahren, S. Kyle, and W. H. Moir. 2001. Responses of exotic plant species to fires in *Pinus ponderosa* forests in northern Arizona. Journal of Vegetation Science 12:261-268.

Croft, L. K. 2001. Rare plants of eastern Oregon and Washington. Northwest Science 75(Special Issue):149-156.

D'Antonio, C. M. and P. M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Review of Ecology and Systematics 23:63-87.

Davies, K. F., P. Chesson, S. Harrison, B. D. Inouye, B. A. Melbourne, K. J. Rice. 2005. Spatial heterogeneity explains the scale dependence of the native-exotic diversity relationship. Ecology 86:1602-1610.

Dickson, B. G. 2006. Multi-scale response of avian communities to prescribed fire: implications for fuels management and restoration treatments in southwestern ponderosa pine forests. Dissertation, Colorado State University, Fort Collins, CO.

Dickson, B. G., J. W. Prather, Y. Xu, H. M. Hampton, E. N. Aumack, and T. D. Sisk. 2006. Mapping the probability of large fire occurrence in northern Arizona. Landscape Ecology 21:747-761.

Dombeck, M. P., J. E. Williams, and C. A. Wood. 2004. Wildfire policy and public lands: Integrating scientific understanding with social concerns across landscapes. Conservation Biology 18:883-889.

Finney, M. A., C. W. McHugh, and I. C. Grenfell. 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. Canadian Journal of Forest Research 35:1714-1722.

Friederici, P. 2003. The Flagstaff model. Pages 7-25 *in* P. Friederici (ed.). Ecological restoration of southwestern ponderosa pine forests. Island Press, Washington, D.C., USA.

Fulé, P. Z., W. W. Covington, and M. M. Moore. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications 7:895-908.

Fulé, P. Z., A. E. Cocke, T. A. Heinlein, and W. W. Covington. 2004. Effects of an intense prescribed forest fire: is it ecological restoration? Restoration Ecology 12:220-230.

Graham R. T., S. McCaffrey, and T. B. Jain (eds.). 2004. Science basis for changing forest structure to modify wildfire behavior and severity. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-120, Fort Collins, Colorado, USA.

Green, R.H. 1979. Sampling design and statistical methods for environmental biologists. Wiley, New York, NY, USA.

Griffis, K. L., J. A. Crawford, M. R. Wagner, and W. H. Moir. 2001. Understory response. to management treatments in northern Arizona ponderosa pine forests. Forest Ecology and Management 146:239-245.

Hall, K. R. and S. L. Maruca. 2001. Mapping a forest mosaic: a comparison of vegetation and bird distributions using geographic boundary analysis. Plant Ecology 156:105-120.

Hampton, H. H., Y. Xu, J. W. Prather, E. N. Aumack, B. G. Dickson, M. M. Howe, and T. D. Sisk. 2003. Spatial tools for guiding forest restoration and fuel reduction efforts. Proceedings of the 2003 ESRI international user CNFerence, July 7-11, San Diego, CA, USA.

Harrington, M. G. and S. S. Sackett. 1990. Using fire as a management tool in southwestern ponderosa pine. Pages 122-133 *in* J. S. Krammes (ed.). Effects of fire management of southwestern natural resources. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report, RM-GTR-191, Fort Collins, CO, USA.

Harrod, R. J. 2001. The effect of invasive and noxious plants on land management in eastern Oregon and Washington. Northwest Science 75(Special Issue):85-90.

Hobbs, R. J. and L. F. Huenneke. 1992. Disturbance, diversity, and invasion: implications for conservation. Conservation Biology 6:324-337.

Huisinga, K. D., D. C. Laughlin, P. Z. Fulé, J. D. Springer, and C. M. McGlone. 2005. Effects of an intense prescribed fire on understory vegetation in a mixed conifer forest. Journal of the Torrey Botanical Society 132: 590-601.

Kaufmann, M. R., L. S. Huckaby, P. J. Fornwalt, J. M. Stoker, and W. H. Romme. 2003. Using tree recruitment patterns and fire history to guide restoration of an unlogged ponderosa pine/Douglas-fir landscape in the southern Rocky Mountains after a century of fire suppression. Forestry 76: 231-241.

Kerns, B. K., W. G. Thies, and C. G. Niwa. 2006. Season and severity of prescribed burn in ponderosa pine forests: Implications for understory native and exotic plants. Ecoscience 13:44-55.

Knapp, E. E. and J. E. Keeley. 2006. Heterogeneity in fire severity within early season and late season prescribed burns in a mixed-conifer forest. International Journal of Wildland Fire 15:37-45.

Kotliar, N. B., S. Hejl, R. L. Hutto, V. Saab, C. P. Melcher, and M. E. McFadzen. 2002. Effects of fire and post-fire salvage logging on avian communities in conifer dominated forests of the western United States. Pages 49-64 *in* T. L. George and D. S. Dobkin (eds.). Effects of habitat fragmentation on birds in western landscapes: contrasts with paradigms from the eastern United States. Studies in Avian Biology 25.

Legendre, P. and L. Legendre. 1998. Numerical ecology. Second edition. Elsevier, Amsterdam, The Netherlands.

Littell, R. C., G. A. Milliken, W. W. Stroup, and R. D. Wolfinger. 1996. SAS System for Mixed Models. SAS Institute, Inc., Cary, NC, USA.

Longley, P. A., M. F. Goodchild, D. J. Maguire, and D. W. Rhind. 2001. Geographic information systems and science, John Wiley and Sons, Chichester, UK.

Lyon, L. J., M. H. Huff, and J. K. Smith. 2000. Fire effects on fauna at landscape scales. Pages 43-49 *in* J. K. Smith (ed.). Wildland fire in ecosystems: effects of fire on fauna. USDA Forest Service, Rocky Mountain Research Station, General Technical Report, RMRS-GTR-42-Volume 1. Ogden, UT, USA.

Manly, B. F. J. 2005. Multivariate statistical methods: a primer. Third edition. Chapman and Hall, Boca Raton, LA, USA.

McGarigal, K., S. A. Cushman, and S. Stafford. 2000. Multivariate statistics for wildlife and ecology research. Springer-Verlag, New York, NY, USA.

McHugh, C. W. and T. E. Kolb. 2003. Ponderosa pine mortality following fire in northern Arizona. International Journal of Wildland Fire 12:7-22.

Miller, C. and D. L. Urban. 2000. Modeling the effects of fire management alternatives on Sierra Nevada mixed-conifer forests. Ecological Applications 10:85-94.

Moir, W. H., B. W. Geils, M. A. Benoit, and D. Scurlock. 1997. Ecology of southwestern ponderosa pine forests. Pages 3-27 *in* W. M. Block and D. M. Finch (eds.). Songbird ecology in southwestern ponderosa pine forests: a literature review. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-292, Fort Collins, CO, USA.

Mutch, R. W. 1995. Fighting fire with prescribed fire: a return to ecosystem health. Journal of Forestry 92:31-33.

Neter, J., W. Wasserman, and M. H. Kutner.1990. Applied linear statistical models. Third edition. Richard D. Irwin, Homewood, IL, USA.

Noss, R. F., P. Beier, W. W. Covington, R. E. Grumbine, D. B. Lindenmayer, J. W. Prather, F. Schmiegelow, T. D. Sisk, and D. J. Vosick. 2006. Recommendations for integrating restoration ecology and conservation biology in ponderosa pine forests of the southwestern United States. Restoration Ecology 14:4-10.

Pollet, J. and P. N. Omi. 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. International Journal of Wildland Fire 11:1-10.

Saab, V. A., L. Bate, J. Lehmkuhl, B. G. Dickson, S. Story, S. Jentsch, and W. M. Block. In Review. Changes in downed woody material and forest structure after prescribed fire in ponderosa pine forests. RMRS Technical Publication.

Sackett, S. S., S. M. Haase, and M. G. Harrington. 1996. Prescribed burning in southwestern ponderosa pine. Pages 178–186 *in* P. F. Ffolliott, L. F. DeBano, M. B. Baker Jr., G. H. Gottfried, G. Solis-Garza, C. B. Edminister, D. G. Neary, and R. H. Hamre (eds.). Effects of fire on Madrean province ecosystems. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-289, Fort Collins, CO, USA.

Sackett, S. S. and S. M. Haase. 1998. Two case histories for using prescribed fire to restore ponderosa pine ecosystems in northern Arizona. Pages 380-389 *in* T. L. Pruden and L.A. Brennan (eds.). Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology CNFerence Proceedings, Number 20. Tall Timbers Research Station, Tallahassee, FL, USA.

Sakai, A. K., F. W. Allendorf, J. S. Holt, D. M. Lodge, J. Molofsky,, K. A. With, S. Baughman, R. J. Cabin, J. E. Cohen, N. C. Ellstrand, D. E. McCauley, P. O'Neil, I. M. Parker, J. N. Thompson, and S. G. Weller. 2001. The population biology of invasive species. Annual Review of Ecological Systems 32:305-332.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The interaction of fire, fuels, and climate across rocky mountain forests. BioScience 54:661-676.

Sesnie, S. and J. D. Bailey. 2003. Using history to plan the future of old-growth ponderosa pine. Journal of Forestry 101:40-47.

Sieg, C. H., B. Philips, and L. Moser. 2003. Exotic and Noxious Plants. Pages 251-267 *in* P. Frederici (ed.). Ecological restoration of southwestern ponderosa pine forests. Island Press, Washington, D.C., USA.

Sisk, T. D., J. W. Prather, H. M. Hampton, E. N. Aumack , Y. Xu, and B. G. Dickson. 2006. Participatory landscape analysis to guide restoration of ponderosa pine ecosystems in the American Southwest. Landscape and Urban Planning 79:xx-xx.

Stein, B. A., L. S. Kutner, and J. S. Adams, eds. 2000. Precious Heritage: The Status of Biodiversity in the United States. Oxford University Press, New York, New York, USA.

Smucker, K. M., R. L. Hutto, and B. M. Steele. 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecological Applications 15:1535-1549.

Stohlgren, T. J., D. Binkley, G. W. Chong, M. A. Kalkhan, L. D. Schell, K. A. Bull, Y. Otsuki, G. Newman, M. Bashkin, and Y. Son. 1999. Exotic plant species invade hot spots of native plant diversity. Ecological Monographs 69:25-46.

Swetnam, T. W. and J. L. Betancourt. 1990. Fire-southern oscillation relations in the southwestern United States. Science 249:1017-1020.

Swetnam, T. W. and J. L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. Journal of Climate 11:3128-3147.

Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: using the past to manage for the future. Ecological Applications 9:1189-1206.

Tiedemann, A. R., J. O. Klemmedson, and E. L. Bull. 2000. Solution of forest health problems with prescribed fire: are forest productivity and wildlife at risk. Forest Ecology and Management 127:1-18.

Turner, M. G., W. H. Romme, and R. H. Gardner. 1999. Prefire heterogeneity, fire severity, and early postfire plant reestablishment in subalpine forests of Yellowstone National Park, Wyoming. International Journal of Wildland Fire 9:21-36.

Turner, M. G., R. H. Gardner, and R. V. O'Neill. 2001. Landscape ecology in theory and practice. Springer-Verlag, New York, NY, USA.

Turner, M. G., W. H. Romme, and D. B. Tinker. 2003. Surprises and lessons from the 1988 Yellowstone fires. Frontiers in Ecology and the Environment 1:351-358.

Wagner, M. R., W. M. Block, B. W. Geils, and K. F. Wenger. 2000. Restoration ecology: a new forest management paradigm, or another merit badge for foresters? Journal of Forestry 98: 22-27.

Wilcove, D. S. and L. Y. Chen. 1998. Management costs for endangered species. Conservation Biology 12:1405-1407.

Wilcove, D. S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States: assessing the relative importance of habitat destruction, alien species, pollution, overexploitation, and disease. BioScience 48:607-615.

Xu, Y., J. W. Prather, H. M. Hampton, E. N. Aumack, B. G. Dickson, and T. D. Sisk. 2006. Advanced exploratory data analysis for mapping regional canopy cover. Photogrammetric Engineering and Remote Sensing 72:31-38.

Zhang, T., R. Ramakrishnan, and M. Livny. 1996. BIRCH: an efficient data clustering method for very large databases. Pages 103-114 *in* Proceedings of the ACM SIGMOD CNFerence on management of data, Montreal, CA.

Zimmerman, G. T. 2003. Fuels and fire behavior. Pages 126-143 *in* P. Friederici (ed.). Ecological restoration of southwestern ponderosa pine forests. Island Press, Washington, D.C., USA.

Table 1. Site descriptions and summary statistics for units occurring on the two study sites used to measure the response of exotic plants to prescribed fire in Arizona, 2003-2004. "PF" or "C" next to unit name indicates prescribed fire treatment unit or control unit, respectively. Sites were located on the Kaibab National Forest (KNF) and the Coconino National Forest (CNF).

Unit name	Area (ha)	Mean elevation (m)	SD	Mean slope (deg)	SD	Total # of sampling stations
KNF						
Kendrick (PF)	396	2253.7	5.1	1.98	1.21	40
Moritz (C)	359	2195.1	9.5	1.35	0.70	40
Beale (C)	127	2245.2	4.2	1.70	0.67	10
Total:	882					90
CNF						
Imax (PF)	405	2121.5	23.6	4.08	2.22	40
Buck Mountain (C)	404	2148.7	36.4	6.10	2.38	40
Total:	809					80
Grand Total:	1,691					170

Table 2. Treatment summaries and statistics for each treatment unit used to measure the response of exotic plants to prescribed fire in Arizona, 2003-2004. "NA" indicates information not available. See Table 1 for site acronym definitions.

Ignition date (MM/DD/YY)	Area burned (ha)	Mean temperature (C)	Mean relative humidity (%)	Wind speeds (km/h)	Wind direction	Sky condition	Mean flame lengths (cm)
KNF							
10/27/03	85.4	15.6	60.0	0-11	NA	NA	30-46
11/06/03	101.2	10.0	22.0	0-11	NA	NA	30-46
03/25/04	167.9	15.6	30.0	0-11	NA	NA	30-46
Total:	354.5						
CNF							
09/15/03	101.0	25.9	14.3	NA	NA	NA	15-30
09/18/03	101.0	20.4	16.4	NA	NA	NA	15-30
09/19/03	202.0	23.8	14.7	NA	NA	NA	15-30
Total:	404.0						

Table 3. Factor eigenvalues and rotated sums of squared loadings computed using varimax rotation with Kaiser normalization for the three retained factors used to summarize local-scale forest structure on two sites in Arizona, 2003-2004.

	Initial eigenvalues			Rotated loadings		
Factor	Total	Variance (%)	Cumulative variance (%)	Total	Variance (%)	Cumulative variance (%)
1	3.038	33.78	33.78	2.666	29.63	29.63
2	1.903	21.14	54.90	1.810	20.11	49.74
3	1.057	11.74	66.64	1.522	16.91	66.64
4	0.827	9.19	75.83			
5	0.712	7.91	83.74			
6	0.492	5.46	89.20			
7	0.427	4.74	93.95			
8	0.286	3.18	97.13			
9	0.259	2.87	100.00			

Table 4. Communality estimates and rotated factor loadings for the nine variables and three retained factors used to summarize local-scale forest structure on two sites in Arizona, 2003-2004. **Bold** values indicate variable used to synthesize and describe relative importance to one of the three retained habitat factors.

		Factor		
Structure variable	Communality	1	2	3
Tree density	0.691	**0.727**	-0.173	0.364
Basal area	0.602	0.431	0.038	**0.644**
Mean tree height	0.828	0.200	**0.862**	0.212
SD tree height	0.397	**0.560**	0.097	0.273
PIPO dominance	0.813	-0.223	**0.871**	0.060
Large tree richness	0.724	**0.683**	-0.417	0.289
Large snag sum	0.480	**0.685**	-0.073	0.076
Large snag decay	0.754	**-0.783**	-0.217	0.306
Canopy cover	0.709	-0.002	0.203	**0.817**
Cumulative variance (%)[1]		29.63	49.74	66.64

[1] Computed using rotated loadings. See Table 2.3.

Table 5. Summary statistics for canopy cover (%), basal area (m^2/ha), and tree density (trees/ha) used to quantify the PatchType variable on the two study sites in Arizona, 2003-2004.

	Total # of sampling stations		Canopy cover[1]		Basal area[1]		Tree density[1]	
PatchType	CNF	KNF	Mean (SD)	Range	Mean (SD)	Range	Mean (SD)	Range
Open	21	37	26.1 (18.8)	0–83	10.8 (7.0)	0–40	155.4 (116.4)	0–1279
Less dense	56	36	49.2 (14.3)	0–92	21.0 (6.1)	0–41	342.7 (154.7)	0–1358
More dense	3	17	66.7 (13.9)	0–98	24.3 (4.8)	0–45	624.9 (281.1)	0–1835

[1] Computed using four southwestern study sites.

Table 6. Fire severity cluster centroid statistics for all (n = 112 of 134) treatment unit sampling stations where five tree damage variables were measured on two sites in Arizona. Low (L), moderate (M), and heavy (H) bole char severity categories are summarized by the frequency (N) and proportion of sampling stations in each category. Metrics were computed using four southwestern study sites.

Fire severity	Crown scorch (proportion)		Crown consumption (proportion)		Maximum bole scorch height (m)		Black bole circumference (proportion)		Bole char severity					
									(L)		(M)		(H)	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	N	%	N	%	N	%
Low	0.033	0.048	0.000	0.000	1.65	2.85	0.477	0.317	23	100.0	0	0.0	0	0.0
Moderate	0.148	0.206	0.003	0.013	3.04	2.04	0.737	0.258	0	0.0	69	100.0	0	0.0
High	0.364	0.213	0.084	0.223	6.95	6.33	0.890	0.126	0	0.0	3	15.0	17	85.0
Combined	0.163	0.212	0.017	0.098	3.45	3.74	0.711	0.285						

Table 7. Significance test results for five tree damage variables used to describe the three fire severity clusters on two sites in Arizona. **Bold** text indicates tree damage variable was statistically significant ($P \leq 0.05$) with respect to mean value in two-step cluster analysis. Metrics were computed using four southwestern study sites.

Fire severity	*t*-crit (+/-)	Scorched foliage		Consumed foliage		Maximum bole scorch height		% Black bole circumference		χ^2-crit	Bole char severity	
		t	*P*	*t*	*P*	*t*	*P*	*t*	*P*		χ^2	*P*
Low	2.72	**-12.93**	<0.001	--	--	**-3.03**	0.006	**-3.55**	0.002	5.99	**75.90**	<0.001
Moderate	2.57	-0.60	0.551	**-8.95**	<0.001	-1.69	0.096	0.85	0.398	5.99	**38.33**	<0.001
High	2.76	**4.23**	<0.001	1.34	0.098	**2.47**	0.023	**6.37**	<0.001	5.99	**89.00**	<0.001

Table 8. Models of the response of exotic plants to habitat covariates on two sites in Arizona, 2003-2004. Only those models with Akaike weights (w_i) ≥ 0.02, plus global (fully parameterized) and intercept-only models, computed from all possible subsets ($n = 256$) are shown. Models are arranged from lowest to highest ΔAICc value. "*K*" = total number of model parameters.

Model	*K*	AICc	ΔAICc	w_i
Treatunit + Elev	5	530.67	0.00	0.038
FAC_1	4	531.33	0.66	0.028
Treatunit + FAC_1	5	531.39	0.72	0.026
Intercept only	3	532.69	2.02	0.014
Global	11	540.87	10.20	0.000

Table 9. Cumulative Akaike weights (w_+) for assessing the relative importance of habitat covariates used to predict the point-level response ($\hat{\Delta}^*_j$) of exotic plant richness to prescribed fire on two sites in Arizona. Weights for each covariate were computed by summing the Akaike weights across all (n = 256) possible models in which that variable occurred.

Treatment	Elevation	Slope	PatchType	PatchRich	FAC_1	FAC_2	FAC_3
0.549	0.457	0.333	0.346	0.329	0.453	0.276	0.257

Table 10. Model-averaged regression coefficients and unconditional variance estimates (SE) for habitat covariates used to predict the point-level response ($\hat{\Delta}^*_j$) of exotic plant richness to prescribed fire on two sites in Arizona. Estimates for each covariate were derived using all (n = 256) possible model subsets.

Treatment	Elevation	Slope	PatchType	PatchRich	FAC_1	FAC_2	FAC_3
-0.185 (0.231)	-0.047 (0.075)	0.279 (0.083)	-0.059 (0.136)	0.050 (0.127)	0.074 (0.118)	0.003 (0.054)	0.009 (0.075)

Table 11. Regression coefficients for the point-level response ($\hat{\Delta}_j$) of exotic plant richness to four levels of fire severity on two study sites in Arizona.

	PYNU	
Severity level	β	SE
Unburned	0.064	0.132
Low	0.105	0.293
Moderate	-0.023	0.194
High	-0.001	0.736

Table 12. Values of Akaike's Information Criterion (corrected for small sample size: AICc) and AICc differences (ΔAICc) for three model forms and an intercept only model used to predict the point-level response ($\hat{\Delta}^*_j$) of exotic plant species richness to four levels of fire severity on two study sites in Arizona. Models used for inference determined by values of ΔAICc ≤ 2.

	AICc	ΔAICc
Linear	538.8	6.10
Log normal	534.7	2.00
Polynomial	536.7	4.00
Intercept only	532.7	0.00

Table 13. Models of the response of exotic plants to forest structure attributes at two spatial scales on two sites in Arizona, 2003-2004. Only those models with Akaike weights (w_i) ≥ 0.02, plus global (fully parameterized) and intercept-only models, computed from all possible subsets ($n = 256$) are shown. Models are arranged from lowest to highest ΔAICc value. "*K*" = total number of model parameters.

Model	*K*	AICc	ΔAICc	w_i
Canopy(local)	4	531.99	0.00	0.040
Treatunit + Canopy(local)	5	532.26	0.27	0.034
BA(local) + Canopy(local)	5	532.33	0.34	0.033
BA(patch) + Canopy)local)	5	532.36	0.37	0.033
Intercept only	3	532.69	0.70	0.028
Treatunit	4	532.71	0.72	0.028
Treatunit + BA(local) + Canopy(local)	6	532.72	0.73	0.027
Treatunit + BA(patch) + Canopy(local)	6	532.84	0.85	0.026
StemDensity(local)	4	533.35	1.36	0.020
Global	11	540.34	8.35	0.000

Table 14. Cumulative Akaike weights (w_+) for assessing the relative importance of forest structure attributes at two spatial scales used to predict the point-level response ($\hat{\Delta}^*_j$) of exotic plant richness to prescribed fire on two sites in Arizona. Weights for each covariate were computed by summing the Akaike weights across all (n = 256) possible models in which that variable occurred.

Treatment	Canopy(local)	BA(local)	StemDensity(local)	Canopy(patch)	BA(patch)	StemDensity(patch)
0.459	0.595	0.339	0.305	0.267	0.368	0.296

Table 15. Model-averaged regression coefficients and unconditional variance estimates (SE) for forest structure attributes at two spatial scales used to predict the point-level response ($\hat{\Delta}^*_j$) of exotic plant richness to prescribed fire on two sites in Arizona. Estimates for each covariate were derived using all (n = 256) possible model subsets.

Treatment	Canopy(local)	BA(local)	StemDensity(local)	Canopy(patch)	BA(patch)	StemDensity(patch)
-0.125 (0.192)	-0.006 (0.006)	0.004 (0.011)	0.000 (0.001)	-0.004 (0.040)	0.061 (0.140)	-0.018 (0.109)

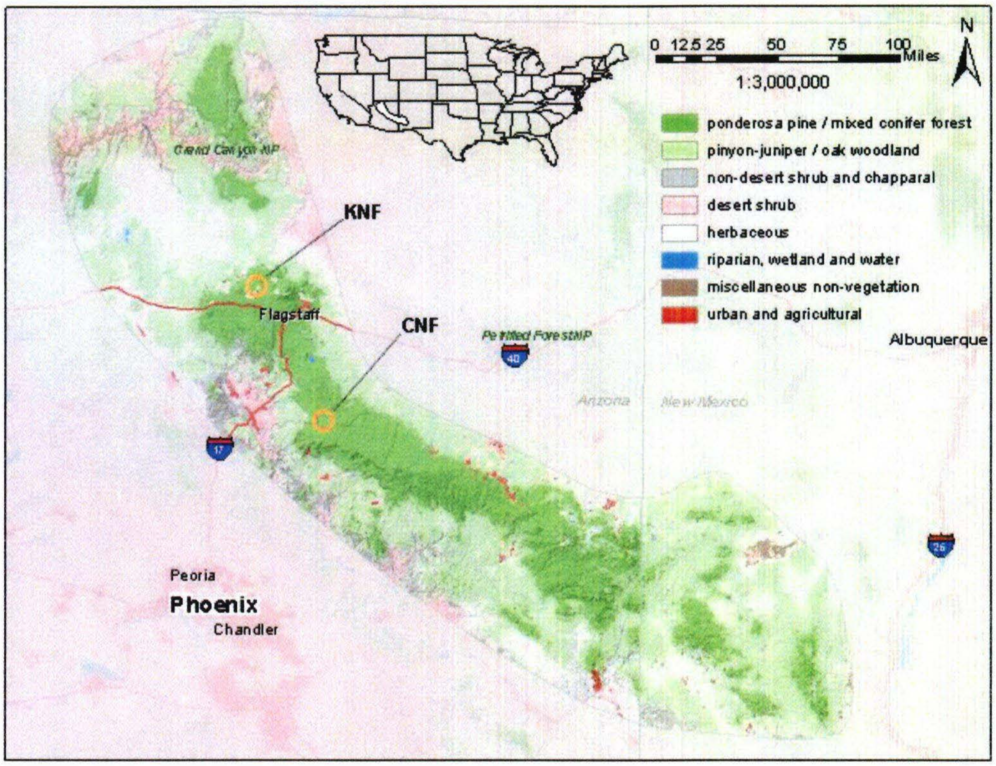
N
0 12.5 25 50 75 100 Miles
1:3,000,000
ponderosa pine / mixed conifer forest
pinyon-juniper / oak woodland
non-desert shrub and chapparal
desert shrub
herbaceous
riparian, wetland and water
miscellaneous non-vegetation
urban and agricultural
Grand Canyon NP
KNF
Flagstaff
CNF
Petrified Forest NP
40
Albuquerque
Arizona
New Mexico
17
25
Peoria
Phoenix
Chandler

Figure 1. Locations of the two study sites (orange circles) in Arizona used to evaluate exotic plant community response to prescribed fire treatments in ponderosa pine forests on the Coconino National Forest (CNF) and Kaibab National Forest (KNF), 2003–2004.

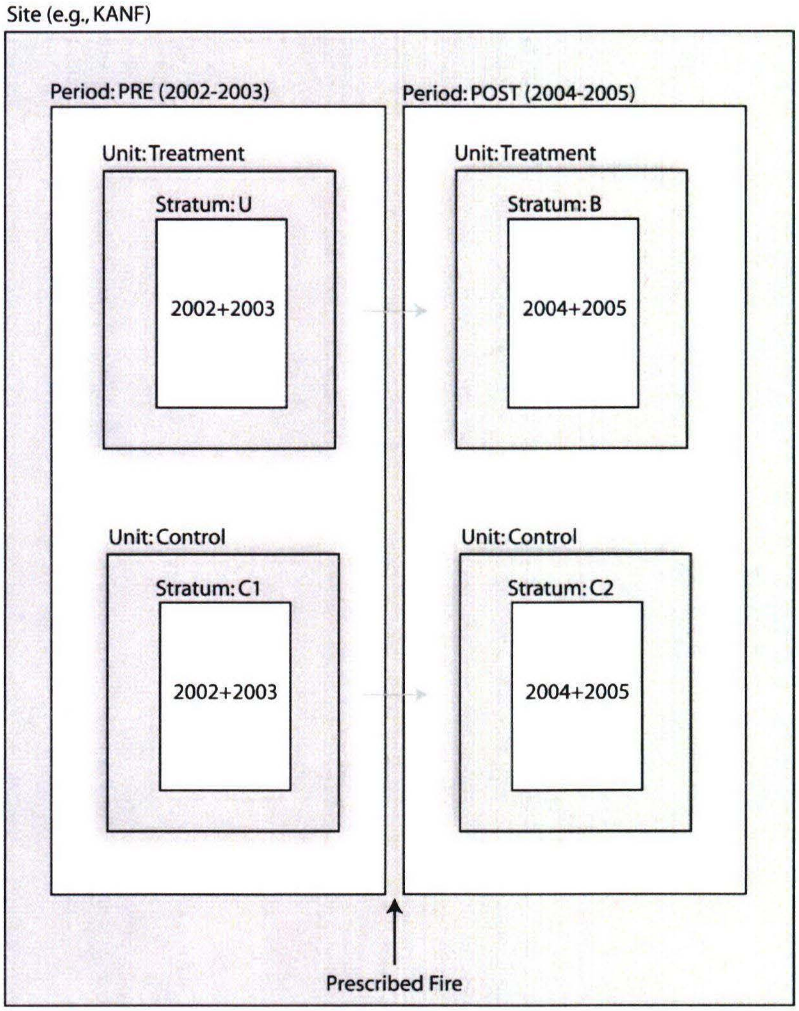
Site (e.g., KANF)
Period: PRE (2002-2003)
Period: POST (2004-2005)
Unit: Treatment
Stratum: U
2002+2003
Unit: Treatment
Stratum: B
2004+2005
Unit: Control
Stratum: C1
2002+2003
Unit: Control
Stratum: C2
2004+2005
Prescribed Fire

Figure 2. Schematic diagram of the Before-After/Control-Impact (BACI) study design used to evaluate exotic plant community response to prescribed fire treatments in ponderosa pine forests on two study sites in Arizona, 2003–2004. "PRE" and "POST" indicate pre- and post-treatment periods, respectively. Stratum levels are indicated by U: unburned treatment unit; C1: pre-treatment control units; B: burned treatment unit; and C2: post-treatment control units.

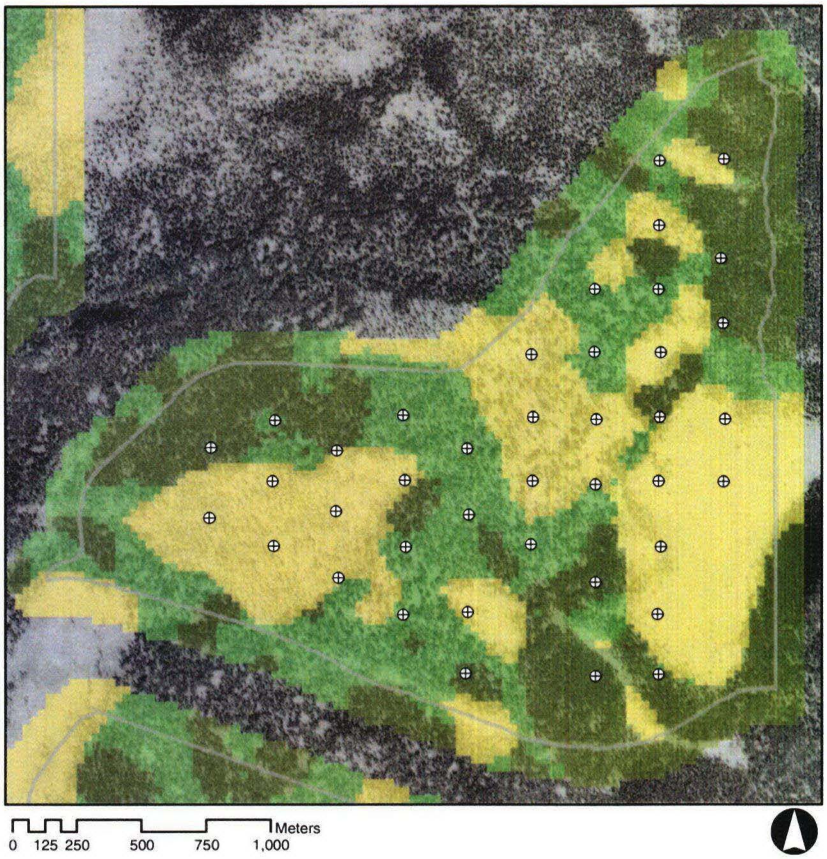

Figure 3. Map detailing the occurrence of three patch types and sampling stations (*n* = 40 points) on the Kendrick prescribed fire-treatment unit (including 100-m buffer), Kaibab National Forest, AZ. Patch types are identified as "open" (yellow), "less dense" (light green), and "more dense" (dark green). Digital orthophoto provided for reference. The southeast corner of the Moritz control unit and northern edge of the Beale control unit are visible.

Appendix A. Summer 2006 progress report for tasks performed in accordance with pre-existing Phase III workplan (01-JV-11221615-186).

During the 2006 field season, the Cooperator, in collaboration with Lindsay Berk from the University of North Carolina, sampled bird and insect communities and measured the effects of prescribed fire treatments on vegetation and forest structure attributes, including monitoring the effects of fire severity on mortality of ponderosa pine trees and repeated measures of vegetation on control units. Presently, data are being transferred from field forms to digital format for analyses. Preliminary results are presented below.

Between 5/29/06-8/18/06, field crews focused data collection efforts on: 1) the distribution and abundance of breeding birds using distance sampling methods; 2) observations of foraging bird behaviors; 3) forest structure attributes at all control units; and 3) understory vegetation structure and diversity, in collaboration with Carolyn Sieg and Jim Fowler.

Avian community sampling

Avian distance sampling protocols were implemented on control and treatment units at all four study sites (ASAZ, COAZ, GINM, and KAAZ). Avian counts were conducted during up to four visits to each of the 278 sampling stations. As a result of an involuntary evacuation due to the Bear wildland fire, sampling stations on the GINM site received only three visits. For the 2006 season, distance sampling efforts resulted in 1,313 individual bird detections on the GINM site, 1,985 detections on the ASAZ site, and 5,675 detections on the KAAZ site. Species richness estimates for breeding birds are 33 on the GINM site, 43 on the ASAZ site and 48 on the KAAZ site. Data entry for the COAZ site is in progress.

Avian foraging behavior was observed and quantified across all treatment and control units on each of the four study sites. Each unit was systematically searched for foraging birds twice during June and July to obtain a sufficient number of independent observations. Foraging observations were recorded for 230 and 119 individual birds on the ASAZ and GINM sites, respectively, including 1,376 foraging behaviors. Data entry for the KAAZ and COAZ sites is in progress.

Vegetation sampling

Forest structure attribute (e.g., stem density and canopy closure) data were collected at 133 sampling stations on all control units using a modification of the Birds and Burns Network vegetation protocols. Currently, field crews are implementing repeated measures of understory vegetation at sampling stations on the KAAZ site.

Appendix B. Project deliverables and accomplishments.

PEER-REVIEWED PUBLICATIONS

Hampton, H. M., E. N. Aumack, J. W. Prather, B. G. Dickson, Y. Xu, and T. D. Sisk. In Press. Development and transfer of spatial tools based on landscape ecology principles: supporting public participation in forest restoration planning in the southwestern U.S. Chapter 7 *in* A. Perera, L. Buse, and T. Crow eds., Forest landscape ecology: transferring knowledge to practice. Springer, New York, New York, USA.

Saab, V., L. Bate, J. Lehmkuhl, B. G. Dickson, S. Story, S. Jentsch, and W. Block. In Press. Changes in downed woody material and forest structure after prescribed fire in ponderosa pine forests. RMRS General Technical Publication.

Sisk, T. D., J. W. Prather, H. M. Hampton, E. N. Aumack , Y. Xu, and B. G. Dickson. In Press. Participatory landscape analysis to guide restoration of ponderosa pine ecosystems in the American Southwest. Landscape and Urban Planning.

Dickson, B. G. 2006. Multi-scale response of avian communities to prescribed fire: implications for fuels management and restoration treatments in southwestern ponderosa pine forests. Dissertation, Colorado State University, Fort Collins, CO.

Dickson, B. G., J. W. Prather, Y. Xu, H. M. Hampton, E. N. Aumack, and T. D. Sisk. 2006. Mapping the probability of large fire occurrence in northern Arizona, USA. Landscape Ecology, 21(6):747-761.

Prather, J. W., N. L. Dodd, B. G. Dickson, H. M. Hampton, Y. Xu, E. N. Aumack, and T. D. Sisk. 2006. Landscape models to predict the influence of forest structure on tassel-eared squirrel populations. Journal of Wildlife Management, 70(3):723-731.

Xu, Y., J. W. Prather, H. M. Hampton, E. N. Aumack, B. G. Dickson, and T. D. Sisk. 2006. Advanced exploratory data analysis for mapping regional canopy cover. Photogrammetric Engineering and Remote Sensing, 72:31-38.

Hampton, H. M., E. N. Aumack, J. W. Prather, Y. Xu, B. G. Dickson, and T. D. Sisk. 2005. A spatial decision support system for forest restoration. Chapter 4 *in* C. Van Riper, III and D. Mattson, eds., The Colorado Plateau *II*: biophysical, socioeconomic, and cultural research. The University of Arizona Press, Tucson, AZ.

Prather, J. W., H. M. Hampton, Y. Xu, B. G. Dickson, N. L. Dodd, E. N. Aumack, and T. D. Sisk. 2005. Modeling the effects of forest restoration treatments on sensitive wildlife taxa: a GIS-based approach. Chapter 5 *in* C. Van Riper, III and D. Mattson, eds., The Colorado Plateau *II*: biophysical, socioeconomic, and cultural research. The University of Arizona Press, Tucson, AZ.

Sisk, T. D., H. M. Hampton, J. W. Prather, E. N. Aumack, Y. Xu, B. G. Dickson, M. R. Loeser, T. Munoz-Erickson, and J. Palumbo. 2004. Forest Ecological Restoration Analysis (ForestERA) Project Report, 2002-2004. Center for Environmental Sciences and Education, Northern Arizona University.

Hampton, H. H., Y. Xu, J. W. Prather, E. N. Aumack, B. G. Dickson, M. M. Howe, and T. D. Sisk. 2003. Spatial tools for guiding forest restoration and fuel reduction efforts. Proceedings of the 2003 ESRI International User Conference, San Diego, CA, July 7-11.

Dickson, B. G., H. M. Hampton, J. W. Prather, Y. Xu, J. D. Bailey, W. H. Romme, and T. D. Sisk. In Prep. Predicting the effects of landscape-scale restoration treatments and forest management on southwestern ponderosa pine forest structure. Forest Ecology and Management.

Dickson, B. G., B. R. Noon, S. Jentsch, C. H. Flather, and W. M. Block. In Prep. Modeling the multi-scale response of avian communities to prescribed fire: implications for fuels management and restoration treatments in the Southwest. Ecological Monographs.

Xu, Y., J. W. Prather, H. M. Hampton, B.G. Dickson, and T. D. Sisk. In Prep. Analysis of bias and variance in forest structural mapping using ETM imagery and ground plot data. Proceedings of the American Society for Photogrammetry & Remote Sensing Annual Conference, Baltimore, Maryland, March 7-11, 2005.

PROFESSIONAL PRESENTATIONS

Breece, C. R., T. E. Kolb, K. M. Clancy, J. D. McMillin, and B. G. Dickson. 2006. Effects of prescribed fire on tree mortality and bark beetles in southwestern ponderosa pine forests. Ecological Restoration Institute National Conference on Conserving and Restoring Frequent Fire Landscapes of the West: Linking Science, Collaboration and Practice, Flagstaff, AZ, Oct. 24-26.

Hampton, H. M., B. Piehl, B. G. Dickson, Y. Xu, J. Rundall, and T. D. Sisk. 2006. Predicting the potential for erosion and sedimentation following stand-replacing wildfire at the lanscape-scale in Southwest forest and woodland ecosystems. Ecological Restoration Institute National Conference on Conserving and Restoring Frequent Fire Landscapes of the West: Linking Science, Collaboration and Practice, Flagstaff, AZ, Oct. 24-26.

Hampton, H. M., J. B. Abrams, J. W. Prather, B. G. Dickson, Y. Xu, and T. D. Sisk. 2006. Design and transfer of spatial tools for forest restoration planning. 21st Annual Symposium of the US Regional Chapter of the International Association of Landscape Ecology, San Diego, CA, March 28-April 1.

Saab, V., L. Bate, J. Lehmkuhl, B. G. Dickson, S. Story, S. Jentsch, and W. Block. 2006. Effects of prescribed fire on fuels and forest structure in western ponderosa pine forests. 1st Fire Behavior and Fuels Conference, Portland, OR, March 28-30.

Breece, C. R., T. E. Kolb, K. M. Clancy, J. D. McMillin, and B. G. Dickson. 2005. Effects of prescribed fire on bark beetle colonization in Southwest ponderosa pine forests. 8th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 7-10.

Dickson, B. G., H. M. Hampton, J. W. Prather, T. D. Sisk, and Y. Xu. 2005. Integrated spatial models inform landscape-level planning for forest restoration and treatments. Invited. 8th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 7-10.

Gwinn, N. R., B. G. Dickson, S. Jentsch, and W. M. Block. 2005. Woodpecker response to prescribed fire in southwestern ponderosa pine forests; consideration of the indicator concept. 8th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 7-10.

Prather, J. W., B. G. Dickson, Y. Xu, T. D. Sisk, and H. M. Hampton. 2005. The Forest Ecosystem Restoration Analysis Project: methods for the assessment of fire threat in southwestern coniferous forests. The Nature Conservancy Fire Learning Network, Silver City, New Mexico, February 16.

Reed, S., T. D. Sisk, W. M. Block, and B. G. Dickson. 2005. Avian community responses to forest thinning and prescribed surface fire, alone and in combination. 8th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 7-10.

Dickson, B. G. 2004. Information theory and multi-model inference. Invited seminar. Center for Environmental Science and Education, Northern Arizona University, Flagstaff, AZ, October 21.

Hampton, H. M., E. N. Aumack, J. W. Prather, Y. Xu, B. G. Dickson, and T. D. Sisk. 2004. Western Mogollon Plateau Adaptive Landscape Assessment. USDA Forest Service Southwest Region 'Assessment of Assessments' Workshop. Taos, New Mexico, November 5.

Hampton, H. M., J. W. Prather, Y. Xu, E. N. Aumack, B. G. Dickson, and T. D. Sisk. 2004. Spatial tools supporting public participation in Planning forest restoration. Annual symposium of the U.S. Regional Association of the International Association of Landscape Ecology, Las Vegas, NV, March 30-April 2.

Prather, J., W., E. N. Aumack, T. D. Sisk, H. M. Hampton, B. G. Dickson, and Y. Xu. 2004. An assessment of various definitions of the Wildland-Urban Interface (WUI): implications for management in a southwestern landscape. Ecological Society of America Annual Meeting, Portland, OR, August 1-6.

Prather, J. W., H. M. Hampton, B. G. Dickson, Y. Xu, E. N. Aumack, and T. D. Sisk. 2004. Effects of restoration treatments on fire behavior and biodiversity: can fire hazard be reduced while minimizing impacts to wildlife? Annual symposium of the U.S. Regional Association of the International Association of Landscape Ecology, Las Vegas, NV, March 30-April 2.

Sisk, T. D., H. M. Hampton, J. W. Prather, Y. Xu, E. N. Aumack, and B. G. Dickson. 2004. Landscape ecological models of fire, biodiversity, and watersheds to guide ecological restoration of ponderosa pine forests. Ecological Society of America Annual Meeting, Portland, OR, August 1-6.

Sisk, T. D., H. M. Hampton, J. W. Prather, Y. Xu, E. N. Aumack, and B. G. Dickson. 2004. Landscape-level assessment of fire, biodiversity, and watersheds to guide ecological restoration of ponderosa pine forests. International Association of Wildland Fire (IAWF) 2004 Conference, Reno, NV, March 3-4.

Dickson, B. G., J. W. Prather, Y. Xu, S. Jentsch, H. M. Hampton, W. M. Block, and T. D. Sisk. 2003. Modeling multi-scale patterns of avian species occurrence: implications for fuels management and restoration treatments in the Southwest. 7th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 3-7.

Dickson, B. G., H. M. Hampton, J. W. Prather, and T. D. Sisk. 2003. Modeling the effects of prescribed fire treatments as a forest restoration alternative: a landscape-level approach. Southwest Fire Initiative Conference, Flagstaff, AZ, April 29.

Hampton, H. M., E. N. Aumack, J. W. Prather, Y. Xu, B. G. Dickson, and T. D. Sisk. 2003. A spatial decision support system for forest restoration. 7th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 3-7.

Hampton, H. H., Y. Xu, J. W. Prather, E. N. Aumack, B. G. Dickson, M. M. Howe, and T. D. Sisk. 2003. Spatial tools for guiding forest restoration and fuel reduction efforts. ESRI International User Conference, San Diego, CA, July 7-11

Prather, J. W., B. G. Dickson, Y. Xu, H. M. Hampton, E. N. Aumack, and T. D. Sisk. 2003. Modeling the effects of forest restoration treatments on sensitive wildlife taxa: a GIS-based approach. 7th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 3-7.

Prather, J. W., Y. Xu, T. D. Sisk, B. G. Dickson, H. M. Hampton, and E. N. Aumack. 2003. Using remotely-sensed forest structure data to help predict the effects of forest management on birds in ponderosa pine forests of northern Arizona. Cooper Ornithological Society Annual Meeting, Flagstaff, AZ, April 30-May 2.

Prather, J. W., H. M. Hampton, Y. Xu, B. G. Dickson, E. N. Aumack, M. M. Howe, and T. D. Sisk. 2003. Using FlamMap and remotely sensed forest structural data layers to predict fire behavior across large landscapes. Southwest Fire Initiative Conference, Flagstaff, AZ, April 29.

Prather, J. W., E. N. Aumack, H. M. Hampton, Y. Xu, B. G. Dickson, and T. D. Sisk. 2002. Linking forest management practices, fire threat, and wildlife needs to facilitate forest restoration planning in Arizona. Wildlife Society Southwest Chapter Meeting, Flagstaff, AZ, October 23-24.

POSTERS

Breece, C. R., T. E. Kolb, K. M. Clancy, J. D. McMillin, and B. G. Dickson. 2005. Effects of prescribed fire on bark beetle colonization in the Southwest. Western Forest Insect Working Conference, Victoria, British Columbia, Canada, March 28-April 1.

Dickson, B. G., Y. Xu, H. M. Hampton, J. W. Prather, J. A. Palumbo, J. D. Bailey, W. H. Romme, and T. D. Sisk. 2005. Spatial tools for predicting the effects of fuels and restoration treatments on forested landscapes. Southwest meeting of the Society of American Foresters. St. George, UT, May 11-13.

Fowler, J. F., C. Hull Sieg, and B. G. Dickson. 2005. Exotic plant species richness at two ponderosa pine sites in northern Arizona. 8th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 7-10

Prather, J. W., S. S. Rosenstock, B. G. Dickson, K. Griffis-Kyle, W. M. Block, C. L. Chambers, T. D. Sisk, Y. Xu, and H. M. Hampton. 2005. Modeling the influence of forest structure and composition on avian communities in northern Arizona ponderosa pine forests. 123rd meeting of the American Ornithologists Union. Santa Barbara, CA, August 23-27.

Xu, Y., J. W. Prather, H. M. Hampton, B.G. Dickson, and T. D. Sisk. 2005. Analysis of bias and variance in forest structural mapping using ETM imagery and ground plot data. Annual conference of the American Society for Photogrammetry & Remote Sensing, Baltimore, Maryland, March 7-11, 2005.

Fowler, J., C. H. Sieg, N. Wilson, and B. G. Dickson. 2004. Exotic plant species: differences between roadside and forest interior plots. Ecological Society of America Annual Meeting, Portland, OR, August 1-6.

Hampton, H. M., J. W. Prather, Y. Xu, E. N. Aumack, B. G. Dickson, and T. D. Sisk. 2004. Collaborative tools for landscape-level planning of forest management. Ecological Society of America Annual Meeting, Portland, OR, August 1-6.

Xu, Y., J. W. Prather, H. M. Hampton, B. G. Dickson, E. N. Aumack, and T. D. Sisk. 2004. A notable limitation on mapping large-area forest structure using TM imagery and ground plot data. Annual symposium of the U.S. Regional Association of the International Association of Landscape Ecology, Las Vegas, NV, March 30-April 2.

Fowler, J., C. H. Sieg, and B. G. Dickson. 2003. Classification of drought year vegetation in ponderosa pine understory. 7th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 3-7.

Hampton, H. M., E. N. Aumack, Y. Xu, J. W. Prather, B. G. Dickson, M. M. Howe, K. M. Kolanoski, J. M. Rundall, and T. D. Sisk. 2003. The development of integrated spatial models to aid in prioritizing forest treatments at the landscape scale. Southwest Fire Initiative Conference, Flagstaff, AZ, April 29.

Wilson, N., J. Fowler, C. H. Sieg, and B. G. Dickson. 2003. Roadside vs. forest interior: a comparison of exotic plant species richness in ponderosa pine forests. 7th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 3-7.

Xu, Y., J. W. Prather, H. M. Hampton, E. N. Aumack, B. G. Dickson, and T. D. Sisk. 2003. Mapping forest structure using multi-source spatial data for strategic forest management in ponderosa pine ecosystems. 7th Biennial Conference of Research on the Colorado Plateau, Flagstaff, AZ, November 3-7.

Prather, J. W., H. M. Hampton, Y. Xu, E. N. Aumack, B. G. Dickson, and T. D. Sisk. 2002. Analyzing the potential effects of forest management on biodiversity and fire threat in pine forests across Northern Arizona. Ecological Society of America Annual Meeting, Tucson, AZ, August 4-9.